Section 3
Treatment Effectiveness

The following sections discuss the treatment effectiveness of the CWS demonstration in Silver Plume, Colorado. The discussion includes a background section, a review of the demonstration, demonstration methodology, site demonstration results, and demonstration conclusions.

3.1 Background

The Burleigh Tunnel is located approximately 50 miles west of Denver in the Georgetown-Silver Plume mining district (Figure 1). The Georgetown-Silver Plume mining district occupies an area of about 25 square miles surrounding the towns of Silver Plume and Georgetown. In general, the period of significant silver production in the area commenced in 1872, reached a peak in 1894, and gradually declined after. Mining in the district increased briefly during World Wars I and II, when many old mines were reopened and considerable amounts of lead and zinc were mined from old stopes, dumps, and wastes left from the silver mining boom.

The Burleigh Tunnel drains a group of mines on Sherman and Republican mountains. Many of these mines intercept shallow groundwater migrating through fractures in the rock or surface water collected by stopes. The intercepted waters are transported through the mines and are eventually discharged through the Burleigh Tunnel. The Burleigh Tunnel discharge contains elevated levels of zinc, typically between 45 and 65 mg/L. However, greater than normal precipitation during the spring of 1995 mobilized a large amount of zinc and increased zinc concentrations within the drainage to 109 mg/L. Burleigh Tunnel discharge rates are generally between 40 to 60 gpm and increase to 100 to 140 gpm during spring runoff. The elevated levels of zinc and significant flow rates combine to make the Burleigh Tunnel a major source of zinc to Clear Creek. Because of the large amount of zinc being discharged to Clear Creek and the potential impact of the zinc on the Clear Creek fishery, the drainage from the Burleigh Tunnel was included in the Clear Creek/Central City Superfund site.

The elevation of the Burleigh Tunnel is 9,152 feet, and the climate is typical of mountainous alpine regions in Colorado. Summers are short and cool and winters are long and cold. Strong eastward, down-valley winds are typical during the winter months. Winds are lighter during the summer months and occasionally blow westward, up the valley. Snow accumulation during the winter months in the immediate area of the tunnel is usually not significant due to the open, south-facing exposure of the hillside and high winds. Snow accumulation at higher elevations in more sheltered areas is significant, with some snow fields persisting until late summer. The average annual temperature is approximately 43.5 degrees Fahrenheit (°F), with a mean minimum of 31°F and a mean maximum of 55.9°F. The average annual precipitation is 15.14 inches.

3.2 Review of SITE Demonstration

The SITE demonstration was divided into three phases: (1) CWS treatability study; (2) CWS technology demonstration; and (3) site demobilization. These activities are reviewed in the following sections, which also discuss variations from the work plan and the CWS performance during the technology demonstration phase.

3.2.1 Treatability Study

A treatability study was conducted at the Burleigh Tunnel between June 18, 1993, and August 12, 1993. The goal of the treatability study was to show that bacterial sulfate reduction could remove zinc from the low-sulfate mine drainage from the Burleigh Tunnel and to estimate levels of zinc reduction that could be expected by CWS treatment. The treatability study involved the construction, operation, and sampling of two bioreactors. Each bioreactor was filled with a mixture of composted manure (96 percent) and alfalfa hay (4 percent), the same substrate that was to be used in the CWS demonstration treatment cells. Both reactors used an upflow configuration, in which Burleigh Tunnel drainage entered the bioreactors from the bottom and was forced to flow up through the substrate. The small bioreactor was 4 feet tall and 22 inches in diameter and held approximately 60 gallons of compost and water. The large bioreactor was 8 feet tall and 22 inches in diameter and held approximately 130 gallons of compost and water. The lower 6 inches of each bioreactor was filled with gravel to support inlet piping and minimize channeling. Peristaltic pumps were used to establish a flow rate of 20 to 30 milliliters per minute for the small bioreactor and 50 to 60 milliliters per minute for the large bioreactor. The flow rates for the bioreactors were set to provide an estimated hydraulic residence time of 50 to 100 hours.

The results of the treatability study indicated that after 8 weeks of operation, both bioreactors achieved removal efficiencies of 99 percent for zinc and similar efficiencies for cadmium and manganese. Zinc was the major metal of concern for the Burleigh Tunnel drainage. Sorption of metals in the substrate is believed to be the dominant removal process during the first 1 to 2 weeks of bioreactor operation. After this brief period of sorption, biological sulfate reduction apparently became the primary metal removal process in the bioreactors. Results of sulfate-reducing bacteria counts and sulfate and sulfide analyses indicated that a large population of sulfate-reducing microorganisms was active in the system. The results supported the theory that the bacteria reduce sulfate in the water to hydrogen sulfide ions, which react with dissolved metals to produce insoluble metal sulfides. The results indicated that the Burleigh Tunnel drainage contains a sufficient concentration of sulfate to promote metal removal by microbial sulfate reduction. Compost sample results from both bioreactors indicated that the compost accumulated metals and sulfide but did not become a reactive or hazardous waste after 8 weeks of operation.

3.2.2 Technology Demonstration

Site preparation requirements for the CWS demonstration were minimal because of previous mining and treatability study activities. Moreover, the area surrounding the Burleigh Tunnel adit is level and required only minor grading to install the two CWS treatment cells. Construction of the CWS treatment cells and all drainage conveyances was the responsibility of the developer (CDPHE).

The demonstration evaluated two treatment cells that differed only in flow configuration, one upward and the other downward. The demonstration evaluated the ability of each cell to remove zinc and other metals from the Burleigh Tunnel mine drainage without pretreatment. Efforts were made to maintain constant flow rates; however, flow rates did vary. In addition, several events resulted in brief interruptions of flow to the cells. Approximately 12.7 million gallons of water from the Burleigh Tunnel were passively treated by the upflow constructed wetland cell and 11 million gallons by the downflow CWS over the 46-month demonstration. Figure 3 shows the flow rates measured for both wetland cell effluents during the demonstration.

Throughout the demonstration, mine drainage influent and wetlands system effluent samples were collected for analysis of total metals, anions, total suspended solids (TSS), and total organic carbon (TOC). In addition, wetlands substrate samples were collected monthly for sulfate-reducing bacteria analysis and quarterly for analysis of total metals, acid-volatile sulfides (AVS), and toxicity characteristic leaching procedure (TCLP) metals. The substrate samples were analyzed to evaluate the effectiveness of the treatment system in sequestering zinc, to assess the tendency of the substrate to become a hazardous waste, and to estimate the role of sulfate-reducing bacteria within the wetlands substrate.

3.2.3 Operational and Sampling Problems and Variations from the Work Plan

The CWS experienced several operational problems during the demonstration. Some of these problems resulted in changes to the schedule and sampling events. Problems encountered and resolutions effected during the demonstration are described below.

- The upflow cell froze in December 1993 and remained frozen until the middle of February 1994. The cell froze because flow to the cells was interrupted when the dike within the Burleigh Tunnel collapsed. The dike was quickly repaired; however, as a result of the cold conditions and the lack of flow to the cells, the upflow cell froze to a depth of 18 inches. A livestock water heater and a steam cleaner were used to thaw the cell so that flow through the cell could be maintained. The freezing of the upflow cell delayed the start of the demonstration by 1 month. In order to prevent the upflow cell from freezing during the winter of 1995, straw bales were placed on top of the cell to provide insulation from the cold.

- The insulation provided by the straw bales maintained the wetland water temperatures consistent with influent values and the upflow cell effluent piping did not freeze.

- The 1995 spring runoff was exceptionally high, and more flow was channeled to the CWS than the wetlands were designed to handle. More than 20 gpm were flowing through the upflow cell for a 2-week period in early June 1995. CDPHE responded to the flooding by installing a 6-inch bypass pipe to carry overflow from the influent weir around the wetlands Once installed, the bypass allowed flow rates to be returned to 7 gpm for each cell. However, CDPHE had not removed the straw bales insulating the upflow cell before the spring runoff began, and the straw bales became saturated. The weight of the saturated straw compressed the substrate, reducing the flow within the upflow cell to less than 1 gpm. The straw bales were removed from the upflow cell, and flow was restored to the cell within a week.

- In late November 1994, a large block of rock, roughly 10 feet by 10 feet, fell from the hillside and rolled onto a corner of the upflow CWS cell. The rock appeared to have depressed the effluent accumulation network and created a high spot in the piping at the collection point to the effluent weir. The high point in the piping may have resulted in the collection of precipitated metal sulfides in the piping, causing a flow restriction.

- During the summer and fall of 1994 and 1995, the effluent flowrate from the downflow cell could not be maintained at 7 gpm. It was not clear if biological surface growth, chemical precipitation in the cell, or settling and compaction of fine particles in the substrate was responsible for the decreased cell permeability.

- Several substrate sampling techniques were proposed for the demonstration, including polyethylene dipper and sediment core samplers. Both techniques appeared to be equally effective; however, the dippers were determined to be preferable. The dippers were selected because they were inexpensive and could be dedicated to each sampling cell, reducing the number of equipment blank samples required during the demonstration.

3.2.4 Site Demobilization

The demonstration-scale wetland was removed by CDPHE at the end of the demonstration. Wetland removal entailed:

- Removal and disposal of the wetland substrate

  - Filling the wetland cells with site materials
  - Filling or removal of wetland weirs

- The CWS demonstration substrate was not a hazardous material, and potential disposal options included:

  - Disposal at a municipal landfill
  - Disposal in landfill biobeds (compost piles)
  - Mixing with site mining waste rock and soil to provide needed organic matter
  - Reuse in an interim ponded wetland

- The CWS Demonstration substrate was disposed of in a nearby municipal landfill

3.3 Demonstration Methodology

The primary objectives of the CWS technology demonstration were to (1) measure the reduction of zinc in Burleigh Tunnel drainage resulting from the CWS treatment with respect to cell configuration and seasonal variation (temperature); (2) assess the toxicity of the Burleigh Tunnel drainage; (3) characterize the toxicity reduction resulting from treatment of the drainage by the CWS; and (4) estimate toxicity reductions in the stream (Clear Creek) receiving the Burleigh Tunnel drainage. In addition, secondary objectives of the demonstration included:

- Estimating the metal removal capacity (lifetime) of the substrate, including the effect of treatment cell flow configuration. The results of influent and effluent metal analyses, CWS flow rate data, and TCLP metal analysis were compared to substrate metal accumulation estimates to evaluate the removal capacities of each CWS treatment cell. The TCLP metals analysis was used because the substrate could become a hazardous waste before its metal removal capabilities were exhausted. Replacing the substrate before it becomes a hazardous waste was determined to be the most cost-effective solution.

- Estimating the extent to which sulfate-reduction processes within the CWS are responsible for the removal of zinc from the drainage. Substrate was analyzed for sulfate-reducing bacteria and acid-volatile sulfides to estimate the extent to which sulfate-reduction processes are removing zinc from the drainage. The approximate number of sulfate-reducing bacteria was correlated to metal removal efficiencies as part of the determination. In addition, the accumulation of AVS in the substrate was compared to metal loading in the treatment cells to determine trends. Furthermore, the AVS analyses included an analysis of zinc to verify that the metal sulfides accumulating in the CWS were zinc sulfides. Previous investigations suggested that AVS analyses were indicative of metal sulfide accumulation attributed to sulfate-reducing bacteria (Reynolds 1991).

- Evaluating the impact of the CWS effluent on Clear Creek. Clear Creek samples were analyzed for total metals, TSS, total dissolved solids (TDS), TOC, nitrate, and phosphate. Results of the stream analyses were compared to CWS effluent analyses to assess the effect of CWS effluent on Clear Creek. Clear Creek samples were collected upstream and downstream of the CWS outfall.

- Estimating the capital and operating costs of the CWS.

Critical parameters are the data required to meet the primary objectives. The primary critical parameters were influent and effluent analyses for zinc (total), and toxicity testing with fathead minnows (Pimephalus promelas) and water fleas (Ceriodaphnia dubia).

Noncritical parameters are data required to address secondary objectives of the demonstration. Secondary objectives provide useful information to potential technology users but are not critical to evaluate the technology. The noncritical parameters of the CWS demonstration included:

- Total metals, nitrate and phosphate analysis of the Burleigh Tunnel drainage and CWS effluents

- Metal loading, metal accumulation, and TCLP metals in CWS substrate samples

- Sulfate-reducing bacteria counts and AVS accumulation in CWS substrate samples

- Clear Creek samples for total metals, TDS, TSS, TOC, biochemical oxygen demand (BOD), and aquatic toxicity

- Construction, operation, maintenance, substrate disposal, and miscellaneous costs

3.3.1 Testing Approach

In general, the testing approach of the demonstration incorporated the collection and analysis of wetland influent and effluent samples every 2 weeks for a period of 20 months. Monthly sampling was conducted for the remainder of the nearly 4-year demonstration. The effluent zinc results for each sampling event were compared to influent data and a removal efficiency calculated. An initial 2-week interval was selected because it provided for 3 to 7 pore volumes of water to be passed through the CWS, assuming a hydraulic residence time of between 50 and 100 hours. In addition, the 2-week interval was chosen because several factors, such as precipitation or evaporation, could cause variation in the measured concentration of zinc in wetland effluent samples. By increasing the number of influent and effluent water samples, performance trends display better continuity, the effects of weather are reduced, and calculated removal efficiencies are expected to more closely reflect true values. Also, sampling intervals shorter than 2 weeks were not economically feasible considering the length of the demonstration. The initial 20-month schedule was the maximum time allowable for the demonstration. This time frame is allowed because the CWS is a biological technology and performance depended, in part, on primary substances and nutrients within the substrate. By allowing the system to operate for an extended period, results were expected to show a relationship (positive or negative) between declining nutrient concentrations in the substrate and CWS performance.

The frequency of demonstration toxicity testing was limited to every 3 to 4 months due to budget considerations. Essentially, the sample collection and testing schedule was designed to evaluate toxicity reduction during periods of widely different zinc removal (different seasons) and critical periods for the receiving stream.

3.3.2 Sampling, Analysis, and Measurement Procedures

Mine drainage samples were collected from the influent weir, and CWS effluent samples were collected from the effluent weirs. Clear Creek samples were collected above and below the CWS outfall. Influent and effluent samples were analyzed for total recoverable zinc and toxicity (critical analyses), other metals, anions, TDS, TSS, and TOC (effluent only). These samples were collected at the frequency discussed in the previous section.

Two substrate sampling points were located in each cell. Initially, substrate samples were collected monthly for sulfate-reducing bacteria analysis and quarterly for total metals, AVS, and TCLP metals analyses for a period of 20 months. Quarterly and semi-annual sampling was conducted for the remainder of the demonstration. Substrate samples were collected from two locations within each cell, at approximately 1 to 2 feet below the wetland surface.

Mine drainage, wetlands effluent, and substrate were analyzed for critical and noncritical parameters using the methods listed in Table 3.

Field analyses included measurement of pH and conductivity for all aqueous samples, Eh for wetlands effluent samples, and dissolved oxygen for mine drainage

Table 3. CWS Demonstration Summary of Standard Analytical Methods and Procedures

Parameter	Sample Type	Method Number	Method Title	Source
Metals	Aqueous and Substrate	6010A, 6020, 7470	ICP, ICP/MS, or AA	SW-846[1]
Sulfate	Aqueous	300.0	Ion chromatography	MCAWW[2]
Fluoride	Aqueous	9056	Ion chromatography	SW-846
Nitrate/Nitrite	Aqueous	353.2 and 354.1	Various	MCAWW[2]
Chloride	Aqueous	300.0	Ion chromatography	MCAWW[2]
Total and Orthophosphate	Aqueous	365.3	Various	MCAWW
pH	Aqueous	9040	Electrometric	MCAWW
TSS	Aqueous	160.2	Gravimetric	MCAWW
TDS	Aqueous	160.1	Gravimetric	MCAWW
TOC	Aqueous	9060	Various	SW-846
Ammonia	Aqueous	350.1	Various	MCAWW[2]
Alkalinity	Aqueous	310.1	Various	MCAWW[2]
Sulfide	Aqueous	376.2	Various	MCAWW[2]
Aquatic Toxicity	Aqueous	EPA SOPs[3]		EPA[5]
Acid Volatile Sulfide (AVS)	Substrate	EPA Method	Acid volatile sulfide	EPA 1991
Sulfate reducing bacteria count	Substrate	None	Anaerobic deep tube	CSM[3]
Toxicity leaching procedure	Substrate	1311	ICP, ICP-MS or AA	SW-846
Reactive sulfide	Substrate	EPA[4]	Titration	SW-846
Orthophosphate	Substrate	365.3	Various	MCAWW
Sulfate	Substrate	300.0	Various	MCAWW
Physical parameters	Substrate	Various[3]	Various[3]	ASTM
Residence time	Aqueous	ND	ND	ND
pH	Aqueous	SOP[3] 12		Tetra Tech[6]
Temperature	Aqueous	SOP[3] 11		Tetra Tech[6]
Dissolved oxygen	Aqueous	SOP[3] 62		Tetra Tech[6]
Conductivity	Aqueous	SOP[3] 99		Tetra Tech[6]

Notes:

1 Test Methods for Evaluating Solid Wastes, Volumes IA-IC: Laboratory Manual, Physical/Chemical Methods; and Volume II Field Manual. Physical/Chemical Methods, SW-846. 3d Edition. Office of Solid Waste and Emergency Response. U.S. Environmental Protection Agency (EPA). 1986.

2 Methods for Chemical Analysis of Water and Wastes (MCAWW). EPA 600/4-79-020. Environmental Monitoring and Support Laboratory, Cincinnati, Ohio. EPA. 1983 and subsequent EPA - 600/4.

3 The analytical methods selected for the analysis of critical and noncritical parameters, and the rationale used in their selection, are discussed in Section 4.2.

4 Interim Guidance for Reactive Sulfide. Section 7.3.4.2, SW-846.

5 Methods for Measuring the Acute Toxicity of Effluents and Receiving Waters to Freshwater and Marine Organisms. EPA/600/4-90/027F. EPA 1993.

6 These are field measurements made by Tetra Tech.

and Clear Creek samples. All field measurements were made in accordance with standard operating procedures.

3.4 Site Demonstration Results

This section presents the results of the CWS demonstration conducted from January 1994 to November 1997. Initially, aqueous chemistry data for the Burleigh Tunnel mine drainage are presented, followed by the demonstration results for the two CWS cells (Sections 3.4.1 through 3.4.3).

Section 3.4.4 presents data for the receiving stream, Clear Creek, and Sections 3.4.5 and 3.4.6 present toxicity results. Tables summarizing analytical results for the Burleigh Tunnel mine drainage are included in Appendix A. An evaluation of demonstration data quality parameters for critical analyses is contained in Section 4.

The data discussed in this section were generally collected using demonstration sampling and analysis techniques. However, influent and effluent data for much of 1996 were collected and analyzed by the CDPHE laboratory (Analytica, in Broomfield, Colorado). In addition, data was not collected by Tetra Tech or CDPHE for 3 months (September through November) in 1996. Tetra Tech discontinued CWS sampling at the end of its initial SITE contract and the resumption of sampling was slowed by contractual delays.

3.4.1 Burleigh Mine Drainage Chemistry

The Burleigh Tunnel drains a network of interconnected mines on Republican Mountain and Sherman Mountain. Unlike many metal mine drainages, the Burleigh Tunnel effluent has near-neutral pH and carbonate alkalinity of approximately 100 mg/L.

The mine drainage contains high levels of zinc that typically range from 45 to 65 mg/L. However, in May and June 1995, a great deal of spring snow and rain and a rapid thaw combined to increase the amount of runoff entering the mine network drained by the Burleigh Tunnel. At that time, flow from the tunnel increased from 45 gpm to more than 300 gpm, and zinc concentrations increased from 55 mg/L (April 12, 1995) to 109 mg/L (August 8, 1995).

Over the final 2 years of the demonstration, zinc concentrations in Burleigh Tunnel mine drainage were lower in the winter, dropped again in April or May when flow through the mine workings increased, and rapidly increased in summer, remaining high throughout the fall. During this period, Burleigh Tunnel mine drainage zinc concentrations generally remained between 45 and 84 mg/L, with increases to more than 100 mg/L noted during the late summer and fall. Zinc concentrations in Burleigh Tunnel mine drainage between September and November 1996 are assumed to be similar to zinc concentrations measured during the same period in 1995. Figure 4 shows zinc concentrations for the Burleigh Tunnel mine drainage measured during the demonstration.

In addition to zinc, cadmium, lead, nickel, and manganese are also demonstration metals of interest. Cadmium, lead, and nickel readily form sulfides and are expected to be removed by the CWS. Manganese does not form a stable sulfide but was shown to be removed in a short-term treatability study conducted prior to the demonstration (PRC 1993). Cadmium, lead, and nickel levels were generally less than 0.1 mg/L in the Burleigh Tunnel mine drainage. After the high flow event in 1995, cadmium levels increased to concentrations ranging from 0.11 to 0.26 mg/L. Lead and nickel levels were generally much lower than cadmium and did not increase to the same extent after the high flow event.

Anion concentrations also increased during the demonstration. Sulfate concentrations in the Burleigh Tunnel drainage ranged from 279 to 652 mg/L and also increased after the high flow event. Carbonate (total alkalinity) concentrations were measured over a relatively narrow range of 82.4 to 125 mg/L. The highest carbonate concentrations were measured during a 1-month period in June and July 1995, corresponding to the period of highest flow from the Burleigh Tunnel. The simultaneous increases in zinc, sulfate, carbonate, and calcium without an increase in pH suggest these mine drainage constituents originate from mineral dissolution. Calcite ($CaCO3$) is commonly found in hydrothermal vein deposits in association with lead-silver-zinc formations (Correns 1969) and is also reported in the Silver Plume mining district. The high concentration of both zinc and carbonate at near neutral pH suggests the Burleigh Tunnel mine drainage is a combination of waters from multiple sources.

3.4.2 Downflow CWS

The downflow cell was operated for approximately 2½ years during the demonstration. Over this period, the system removed 60 to 95 percent of the zinc contamination from the Burleigh Tunnel mine drainage.

Figure 4 shows zinc concentrations in the Burleigh Tunnel mine drainage (influent), and the effluents of both CWS

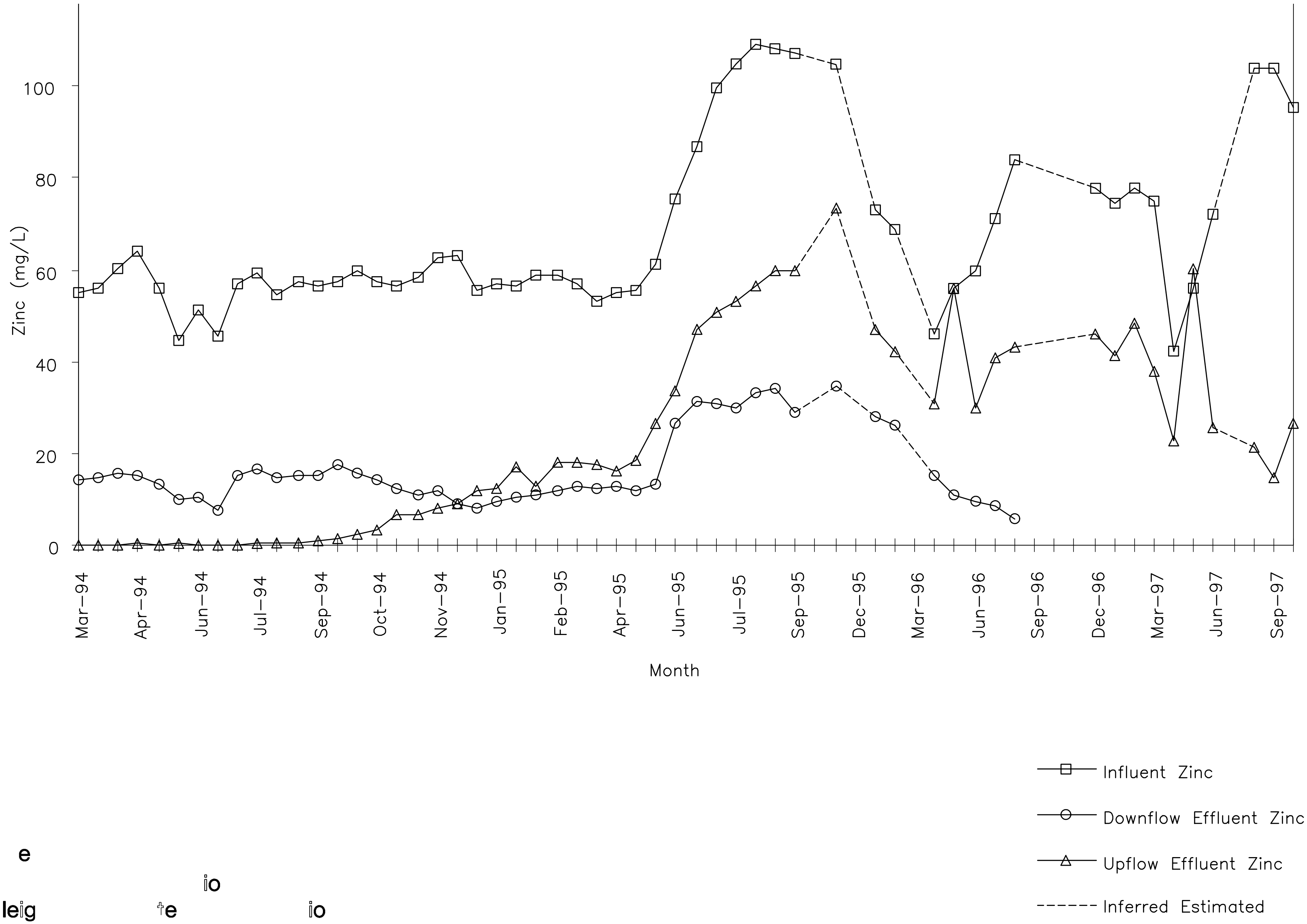

Figure 4. CWS zinc concentrations by month.

cells. During the first year of operation, influent zinc concentrations ranged from 45 to 63 mg/L (average of 57.1 mg/L) and the amount of zinc removed by the downflow cell ranged from 35 to 54 mg/L (average of 44.2 mg/L). Zinc removal efficiency during the first year averaged 77.4 percent. During the second year, zinc levels in mine drainage ranged from 53 to 109 mg/L (average of 83 mg/L) and downflow zinc removal ranged from 41 to 78 mg/L (average of 58 mg/L). Zinc removal efficiency during the second year averaged 70 percent. Over the final 6 months this cell operated, influent zinc levels ranged from 46 to 84 mg/L, while downflow CWS zinc removal ranged from 31 to 78 mg/L. In general, greatest zinc removal corresponded to times with the highest influent zinc concentrations, and the lowest zinc removal was observed during periods of lesser zinc in the mine drainage suggesting metal removal was effected by a physical process.

Although present only in low levels in the influent water, cadmium, lead, and nickel were removed to a great extent by the downflow CWS treatment. Influent cadmium concentrations ranged from 0.071 to 0.10 mg/L, while effluent levels ranged from 0.0007 to 0.003 mg/L during the first year. During the second year, cadmium concentrations increased in the influent, ranging from 0.057 to 0.26 mg/L, and downflow effluent levels ranged from 0.0001 to 0.007 mg/L with few detections. Figure 5 shows cadmium concentrations for the influent and both effluents during the first 2 years of the demonstration Substantial cadmium removal continued over the final 6 months by the downflow cell, with the exception of the April 1996 sample.

Samples were not regularly analyzed for lead or nickel during the demonstration. Figure 6 shows lead concentrations for the influent and both effluents during the first 2 years of the demonstration. During the first year, influent lead concentrations ranged from 0.013 to 0.020 mg/L, while downflow effluent concentrations ranged from 0.00065 to 0.0054 mg/L. Throughout the remainder of 1995, influent levels of lead increased slightly while effluent levels remained very low with few detections.

Nickel was also removed by the downflow cell; however, the extent of removal declined when influent nickel concentrations increased after the high flow event. Nickel levels in the influent ranged from 0.033 to 0.68 mg/L, and downflow effluent ranged from 0.0073 to 0.020 mg/L in the first year. Throughout the remainder of 1995, influent nickel levels ranged from 0.045 to 0.093 mg/L, and downflow effluent levels ranged from 0.014 to 0.040 mg/L.

Manganese concentrations in the mine drainage were initially between 1 to 2 mg/L. Manganese removal by the downflow CWS was low during the demonstration. Figure 7 shows manganese concentrations for the influent and both effluents.

The extended residence time of the influent within the downflow cell substrate caused by low flow rates may be one reason the downflow CWS was effective in removing metals from the mine drainage. Both wetland cells were designed to treat 7 gpm; however, the permeability of the downflow cell declined during the first year of operation, and flow through the cell dropped to 4 gpm particularly during the summer months. Although attempts were made to increase its permeability by fluffing the substrate with compressed air, these procedures resulted in only temporary improvements. Flow through the downflow cell improved during winter months when the substrate froze and contracted from the liner allowing the influent to flow down the sides of the interior cell. Flow through the downflow cell averaged 6.5 gpm during the first year; 5.8 gpm in the second year; and 6 gpm over the final 6 months of operation.

Analytical results for the downflow substrate (Table 4) showed a substantial increase in zinc levels over the period of the demonstration. Substrate zinc levels ranged from a low of 59.7 milligrams per kilogram (mg/kg) to a high of 5,630 mg/kg. Substrate samples were generally collected from between 1 to 2 feet below the surface of the CWS. Downflow substrate samples contained little visible evidence of sulfate reduction and low concentrations of AVS. Sulfate-reducing bacteria counts showed much variability (Figure 8).

After the first 6 months of operation, the downflow cell was removing more zinc from the mine drainage compared with the upflow cell. However, the reason for the greater removal was likely the higher residence time of the mine drainage within the downflow wetland. The increasing residence time was a function of mine drainage flow through the cell, that was generally lower in the summer compared to winter. A reduction of flow from 7 to 5 gpm increases residence time by 19 hours nearly a 40 percent increase. The loss of permeability is believed to be related to the loss of permeability in the downflow cell resulting from biological surface growth, chemical precipitation of

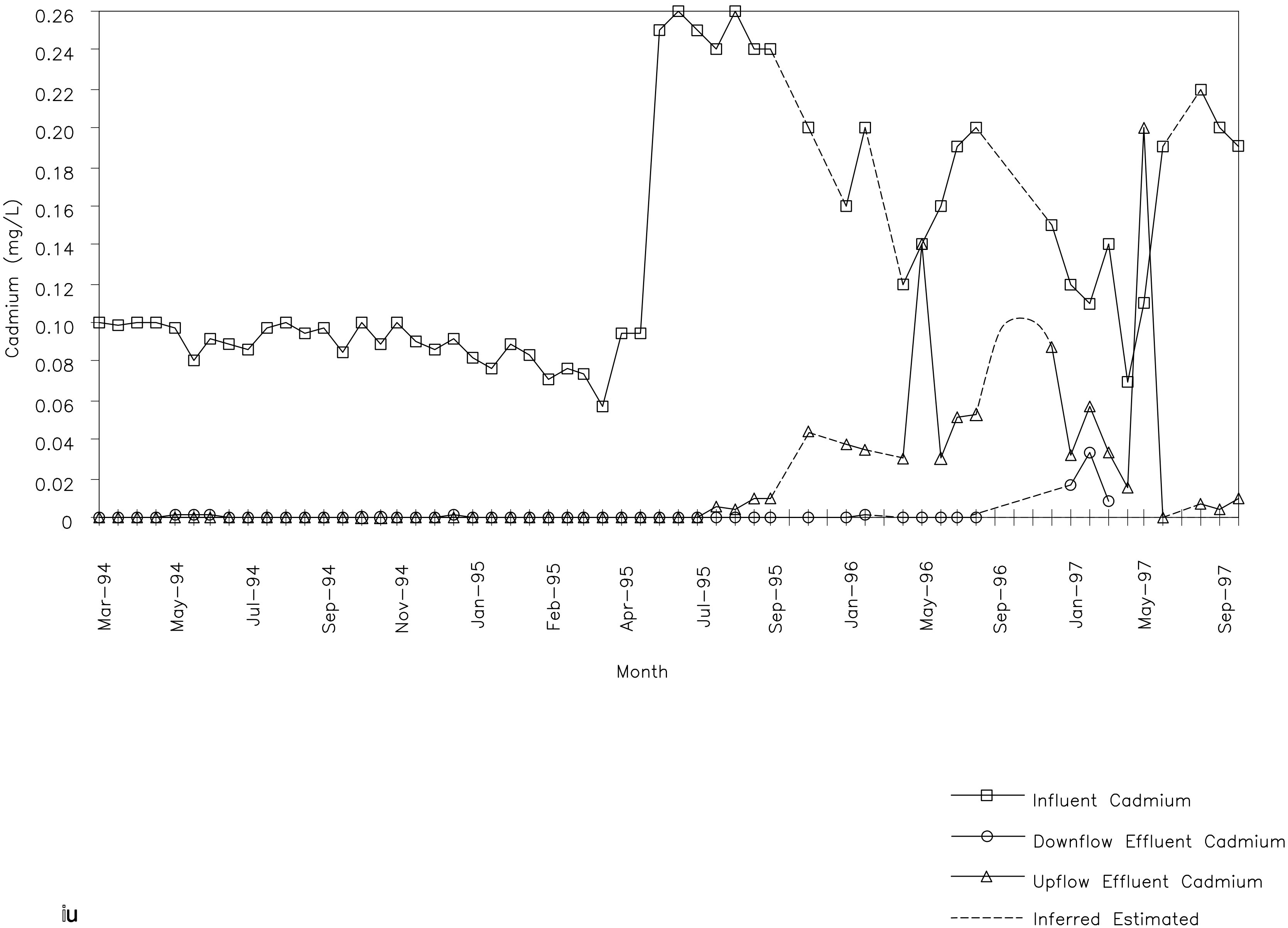

Figure 5. CWS cadmium concentrations by month.

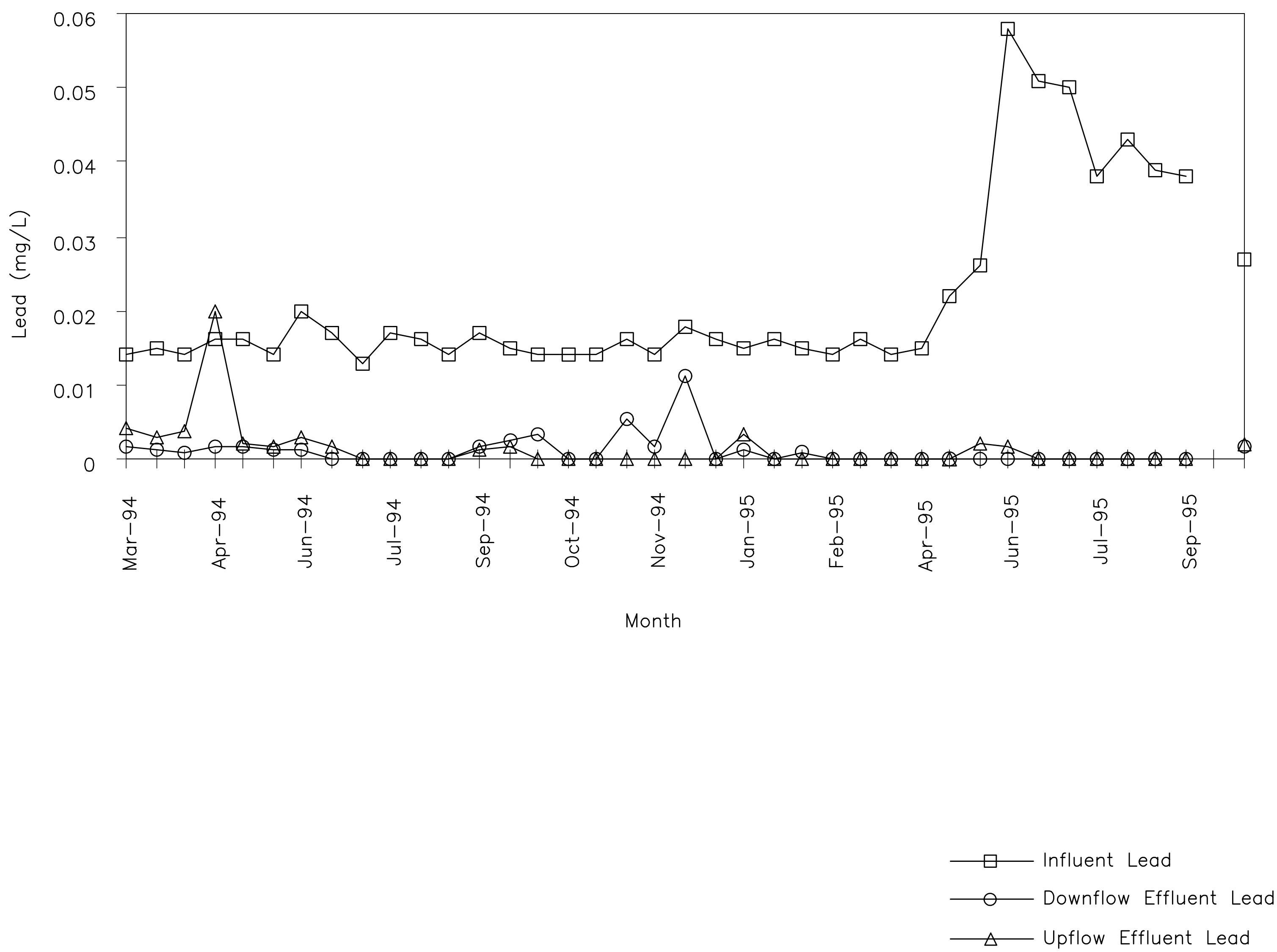

Figure 6. CWS lead concentrations by month.

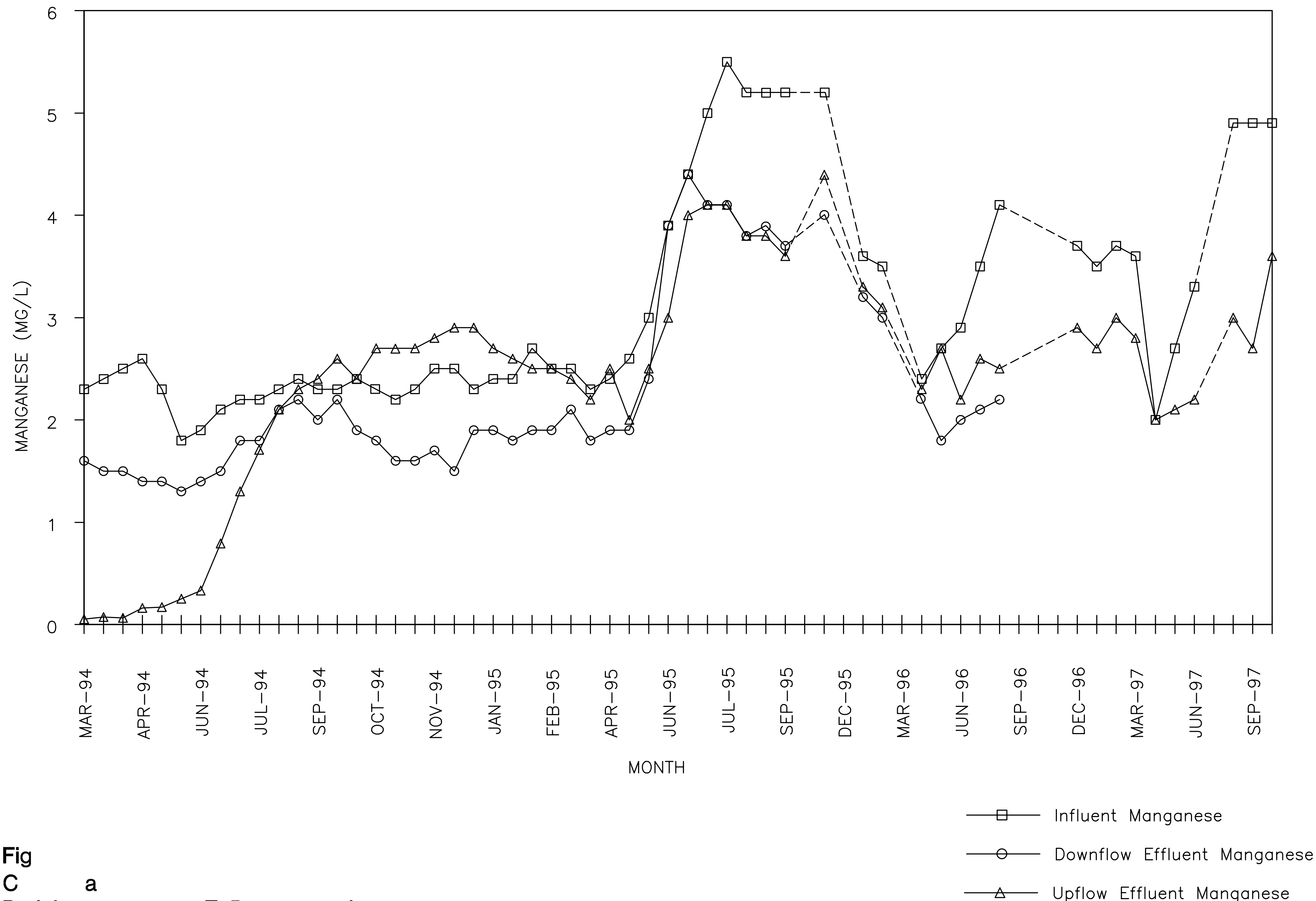

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Figure 7. CWS manganese removed by month.

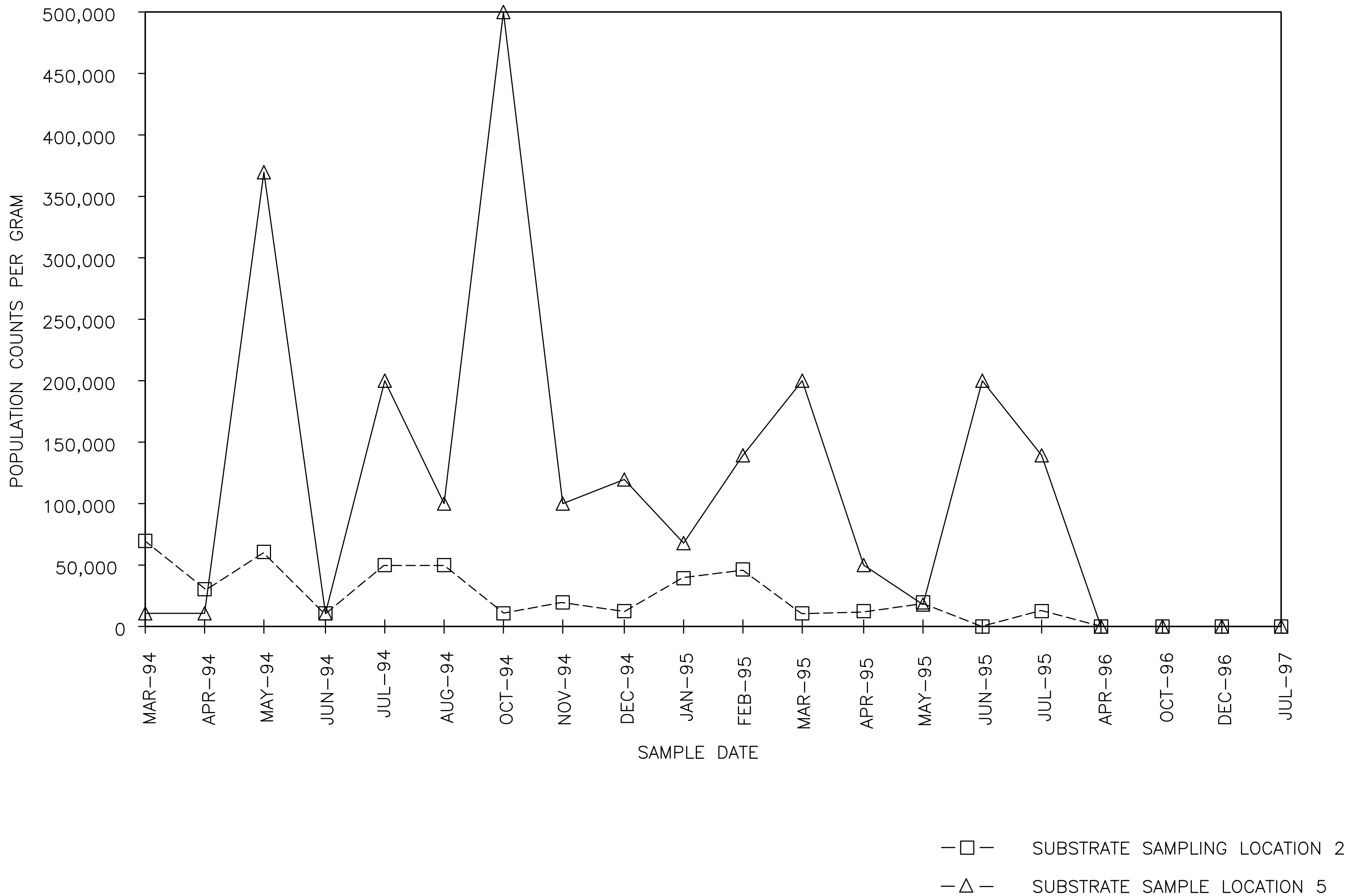

Figure 8. Sulfate-reducing bacteria, downflow CWS substrate.

Table 4. Average Downflow CWS Substrate Results

	Cadmium (mg/kg)	Lead (mg/kg)	Nickel (mg/kg)	Zinc (mg/kg)	Acid Volatile Sulfides (mg/kg)	Sulfate-Reducing Bacteria (count)	Ortho-phosphate (mg/kg)
0-6 months	2.7	18	3.1	1,100	180	8.5×10^4	34
6-12 months	8.0	31	6.1	3,400	120	1.1×10^5	12
12-18 months	23	74	7.0	5,200	460	3.3×10^5	2.6

Notes:

mg/kg Milligram per kilogram
Average Arithmetic Mean
Substrate samples collected from 1-2 feet below wetland surface

zinc compounds, microbial breakdown of the substrate to finer particulates, and the settling of these particles into substrate pore spaces. The increase of flow during winter is believed to result from freezing of the wetland substrate at the edge of the cell causing the substrate to contract from the liner. The contraction allowed ponded water at the surface of the wetland to flow between the frozen substrate and liner to the base of the cell forming a preferential pathway.

Loading is the amount of metals retained by the wetland over time. It is a function of the flowrate through the wetland, the concentration of metals in the mine drainage, and the removal efficiency of the treatment. For this discussion, monthly loading of each wetland was calculated from measured flow rates and simultaneously collected samples of the mine drainage and the wetland effluent. Figure 9 shows the monthly zinc loading to the downflow CWS over the demonstration. The graph indicates that loading was initially high (maximum of 60 kg/month) but dropped as the downflow cell flow rate declined in the Fall of 1994. In winter, loading also increased as flow improved. The greatest loading to the downflow CWS occurred during the high flow event in the late spring and early summer of 1995. After the high flow event, loading in this cell declined dramatically and eventually dropped to less than 5 kg/month in May 1996.

The primary metal removal mechanism active in this cell did not appear to be sulfate reduction. Substrate analyses indicate a significant portion of the zinc removal in this CWS occurred in the upper 1 to 2 feet of substrate, where few AVS or sulfate-reducing bacteria were found. Pockets of sulfide-rich substrate were observed in this CWS cell at depths of 3 to 4 feet below the wetland surface, suggesting some sulfate reduction contributes to metal removal in this wetland. Aqueous geochemical modeling of the mine drainage suggests gypsum is oversaturated; however, visual observations of Burleigh Tunnel mine drainage precipitate and historical mine reports suggest the material is a zinc carbonate, probably smithsonite or hydrozincite.

The following can be concluded from the evaluation of the downflow CWS:

- As tested, the downflow CWS did not retain sufficient permeability to be considered a reasonable long-term treatment option.
- Chemical precipitation (suspected to be mineral carbonate accumulations) may have been the primary metal removal process in this CWS treating Burleigh Tunnel mine drainage.
- A 2-foot substrate depth should be adequate, as most metal removal occurred at between 1 to 2 feet below the wetland surface. A thinner substrate should decrease the flow resistence of the downflow CWS and increase the effectiveness of the system.
- A 2-foot downflow CWS may be a good pretreatment for an upflow CWS treating the Burleigh Tunnel mine drainage allowing some physical precipitation of the zinc.

The concentration of orthophosphate in the substrate also decreased after the high flow event in 1995. The high orthophosphate concentration, measured at the beginning of the demonstration, was 114 mg/kg; the low, 1 to 2 mg/kg, was measured in August 1995.

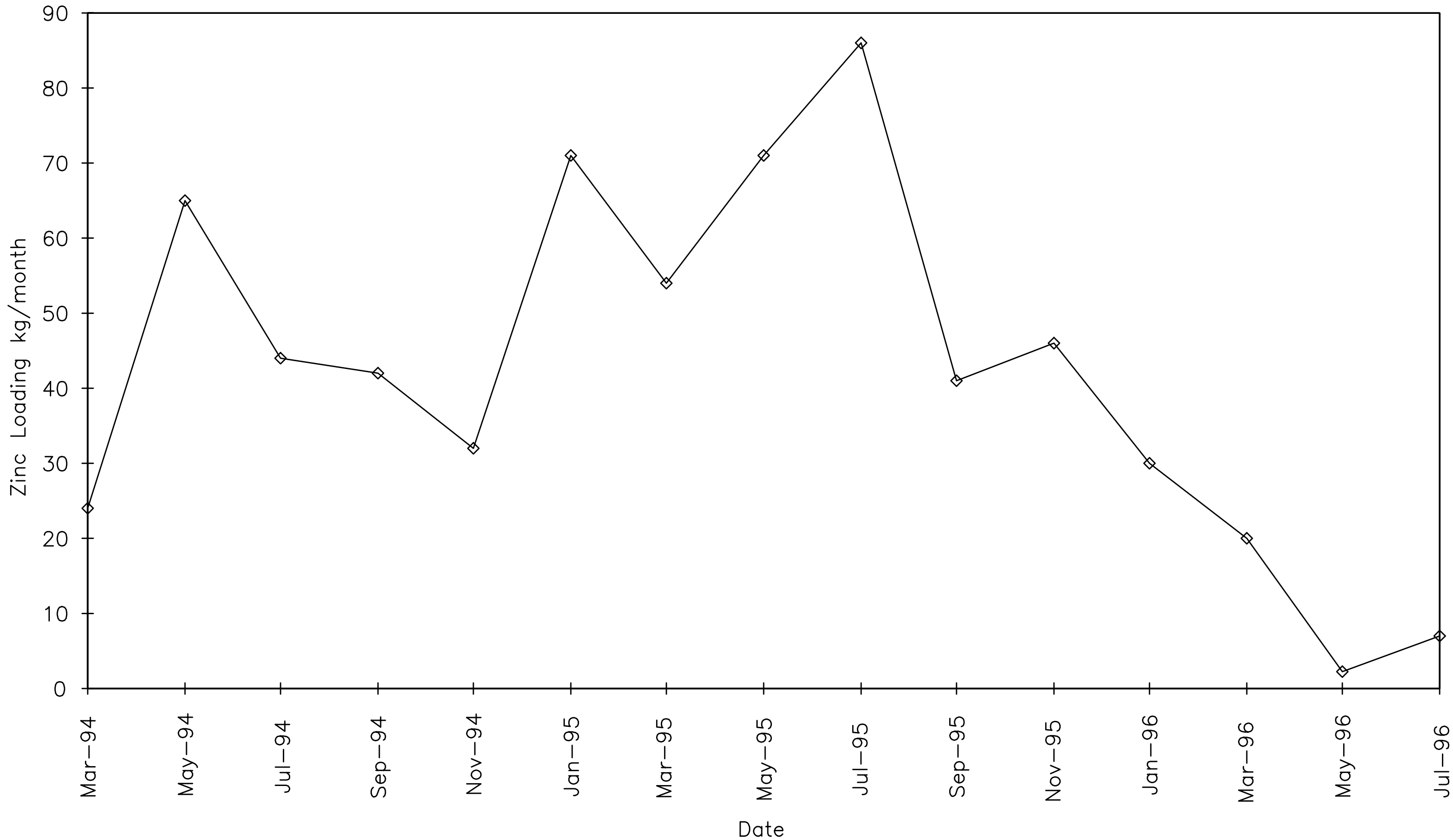

Figure 9. Monthly zinc loading, downflow CWS.

3.4.3 Upflow CWS

The upflow cell was demonstrated for nearly 4 years and, during this period, removed zinc and other metals initially by adsorption, later by sulfate reduction, and eventually by chemical precipitation (presumed). The adsorption period; appeared to last roughly 4 to 5 months as indicated by manganese removal. After the adsorption phase, sulfate reduction appeared to be the primary metal removal process; however, oxidation/reduction (ORP) measurements suggested the activity of the sulfate-reducing bacteria appeared to drop in late fall and through the winter of 1994. Counts of sulfate-reducing bacteria declined coincidentally with the decline in ORP. The drop may have been caused by lower winter temperatures, or an increase in flow through the cell that occurred in September through October 1994, or may result from the use of all the most easily metabolized materials in the compost substrate by the bacteria. During this period, the concentration of zinc in the upflow effluent increased from 3.2 mg/L (October 12, 1994) to 18 mg/L (March 15, 1995).

By May 1995, zinc levels were approaching levels that are inhibitory to sulfate-reducing bacteria at the observed area loading of 250 square feet per gallon. During May and June of that year, the high flow event exposed the wetland sulfate-reducing bacteria to elevated levels of zinc, and the high influent flow probably created aerobic conditions within the cell. The periodic high zinc concentrations observed in influent waters during the summer and fall of 1996 and 1997 likely prevented the sulfate-reducing bacteria from reestablishing activity to previous levels. The flow was halted to the upflow cell in the summer of 1997 for approximately one month for repairs. At that time, much of the water was removed from the cell, allowing wetland sulfate-reducing bacteria an opportunity to become reestablished.

However, there was no indication that the bacteria became re-established during the final 4 to 5 months of the demonstration. One of the repairs involved plugging a short section of the influent piping in the upflow cell. Visible observation of this influent pipe noted a black coating on the inside of approximately 1/16 inch and accumulations of black precipitate nearly filling the holes in the perforated pipe. Overlying the black material in the piping was a layer of cream colored to yellow material up to 1/8 of an inch thick.

Analytical results for influent and effluent samples from the upflow system showed that zinc was nearly completely removed by this system during the first 8 months of the demonstration (Figure 4). After this period, zinc concentrations in the upflow effluent gradually increased from 1.4 mg/L (September 19, 1994) to 18.5 mg/L in the spring of 1995 corresponding to zinc removal efficiencies of 97.6 and 66.8, respectively. In May and June 1995, high flow from the Burleigh Tunnel increased flow through the upflow cell to 20 gpm and zinc concentrations nearly doubled. Over the next 6 months, as flow decreased from the tunnel, influent zinc concentrations rose to a high of 109 mg/L. From May to November 1995, effluent zinc levels increased from 26.7 to 73.6 mg/L. The amount of zinc removed by the upflow cell averaged 41 mg/L (49.3 percent) during the second year.

During the third year of operation, zinc levels in the influent ranged from 56 to 84 mg/L; however, data were not collected between September and November 1996. Zinc concentrations in the upflow effluent over the third year ranged from 30 to 49 mg/L with an average removal of 30 mg/L (39.6 percent). In the final year of operation, zinc influent concentrations ranged from 42 to 104 mg/L and effluent levels ranged from 15 to 60 mg/L with an average removal efficiency of 65.1 percent. Effluent levels were greater in the May 28, 1997 sample (60 mg/L) compared to the influent sample (56 mg/L). Over the final 6 months, the upflow cell removed greater amounts of zinc as flow through the cell decreased. Flow through the upflow cell at this time ranged from 2 to 5 gpm.

Cadmium removal by the upflow cell followed a pattern similar to zinc removal (Figure 5). Initially, cadmium was removed to nondetect levels; however, cadmium concentrations increased two and a half times after the high flow event. After this period, cadmium removal remained high for 4 months but declined in the latter part of 1995 and remained low through 1996 and 1997.

Lead (Figure 6) and nickel were also removed to lower concentrations by the upflow CWS. Influent lead and nickel concentrations were approximately 0.015 mg/L and 0.043 mg/L, respectively. During the first year, lead was removed to nondetect levels and nickel effluent concentrations ranged from 0.0005 to 0.019 mg/L. Unlike zinc and cadmium, lead and nickel concentrations did not increase significantly after the high flow event; however, the removal of both decreased somewhat until flow values through the cell declined in the final months of the demonstration.

Manganese was initially present in the mine drainage at concentrations ranging from 1 to 3 mg/L. Manganese was removed by the upflow cell for the first 4 months of operation but was not removed throughout the remainder of the demonstration.

Analytical results for the upflow substrate showed an increase in zinc levels over the period of the demonstration. Table 5 summarizes mean annual results for selected analysis from upflow cell substrate samples collected during the demonstration. Zinc levels ranged from a low of 40 mg/kg to a high of 4,800 mg/kg. The zinc content is expected to be higher in the removal zone of the upflow cell (deeper in the substrate of the cell). In general, upflow substrate samples were collected approximately 2 feet below the wetland surface, above the removal zone. Counts of sulfate-reducing bacteria in the upflow cell were generally very high between April 1994, through July 1995. However, counts were 1 to 2 orders of magnitude lower in upflow cell samples collected in April 1996 through September 1997. The final substrate sample analyzed for sulfate-reducing bacteria contained approximately 250,000 CFU/gram substrate. Figure 10 shows the results of sulfate-reducing bacteria counts conducted on upflow cell substrate samples collected during the demonstration.

The change from strongly reducing to slightly reducing conditions in the fall of 1994 may have made previously removed metal sulfides less stable within the wetland substrate. Substrate observations in the summer of 1997 indicated there were fewer sulfides present compared to substrate samples collected in 1994 and 1995. If half of the zinc removed in the first year of operation were released over the subsequent 2 years, the resulting zinc increase in the effluent would have been 33 mg/L. The higher zinc concentration measured in the May 28, 1997 effluent sample compared to the corresponding influent sample suggests some previously removed zinc was released.

Between March and December 1994, metals loading to the upflow CWS ranged from 53 to 97 kg/month but dropped to 26 kg/month in February 1995. This drop in loading corresponded with the increase of zinc in the effluent, an increase in ORP, and a decrease in flow rate through the cell. Flow through the cell increased in March and April 1995, leading to higher loading. The maximum loading to the upflow CWS (107 kg/month) occurred in May 1995 during the high flow event. Throughout the remainder of the demonstration, loading to this cell declined as the zinc removal efficiency decreased to 40 to 50 percent; eventually, flow through the cell ended in 1997. Figure 11 shows zinc loading to the upflow CWS over the demonstration.

The effect of the high flow event on the performance of the upflow CWS reveals the major shortcoming of passive systems, the inability to adapt to rapidly changing conditions. In this demonstration, the upflow CWS could not adjust to the increased influx of zinc or the change in environmental conditions.

As several constructed wetlands have successfully treated mine drainage with much higher concentrations of zinc, it may be concluded that the bacteria are somehow able to protect themselves from the high metals concentration. If this mechanism is sulfate reduction, the rate of sulfate

Table 5. Average Upflow CWS Substrate Results

	Cadmium (mg/kg)	Lead (mg/kg)	Nickel (mg/kg)	Zinc (mg/kg)	Acid Volatile Sulfides (mg/kg)	Sulfate-Reducing Bacteria (count)	Ortho-phosphate (mg/kg)
Year 1	0.17	9.9	1.9	40	210	7.2×10^6	55
Year 2	0.18	13	2.0	71	460	3.2×10^6	54
Year 3	5.0	40.0	4.1	1,500	1,300	2.2×10^5	6.3
Year 4	9.6	NR	6.2	4,800	1,000	6.2×10^4	6.9

Notes:

mg/kg Milligram per kilogram
NR Not Reported
Average / Arithmetic Mean
Substrate samples collected from 1-2 feet below wetland surface

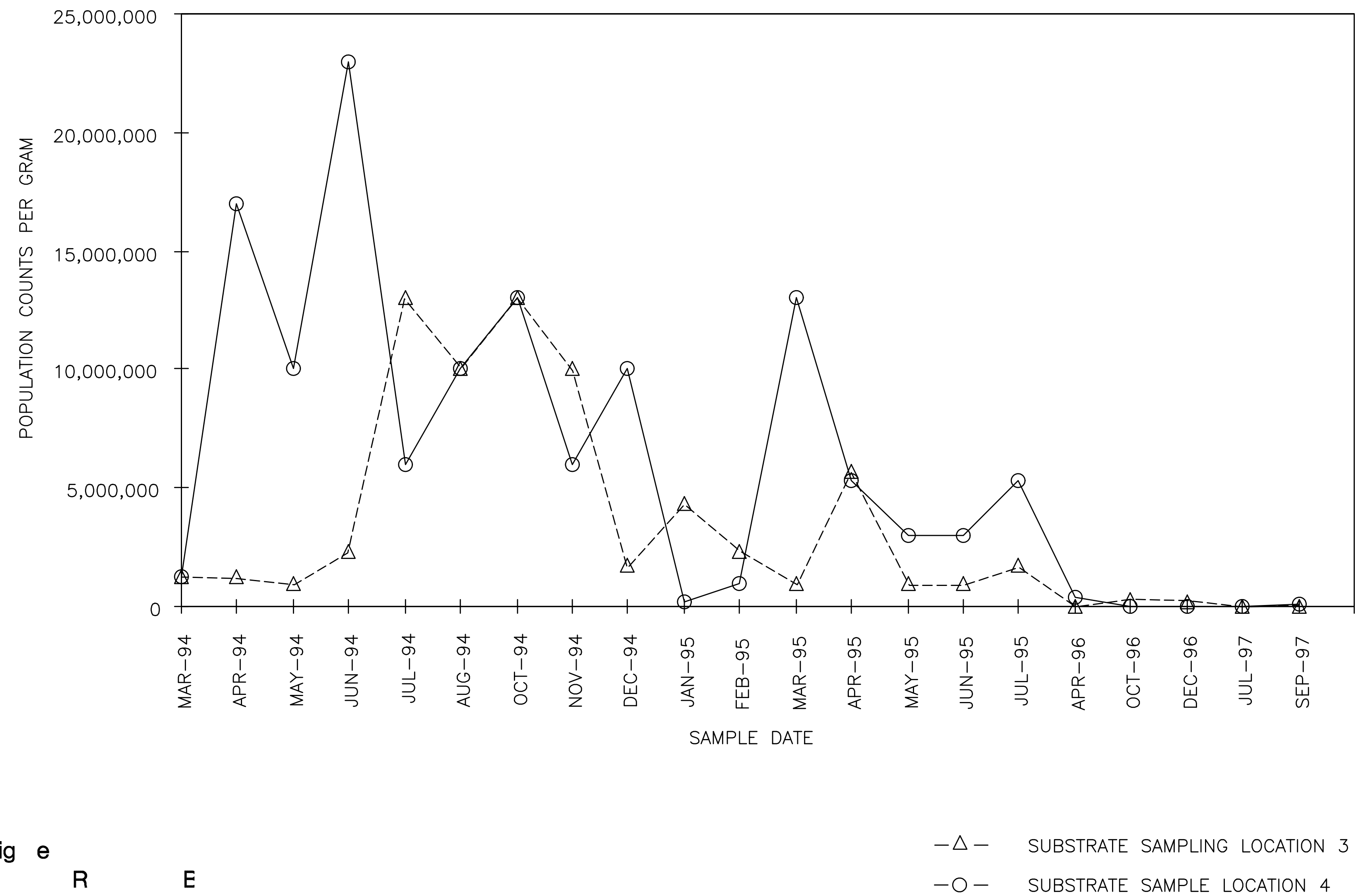

Fig e
S R E

Figure 10. Sulfate-reducing bacteria, upflow CWS substrate.

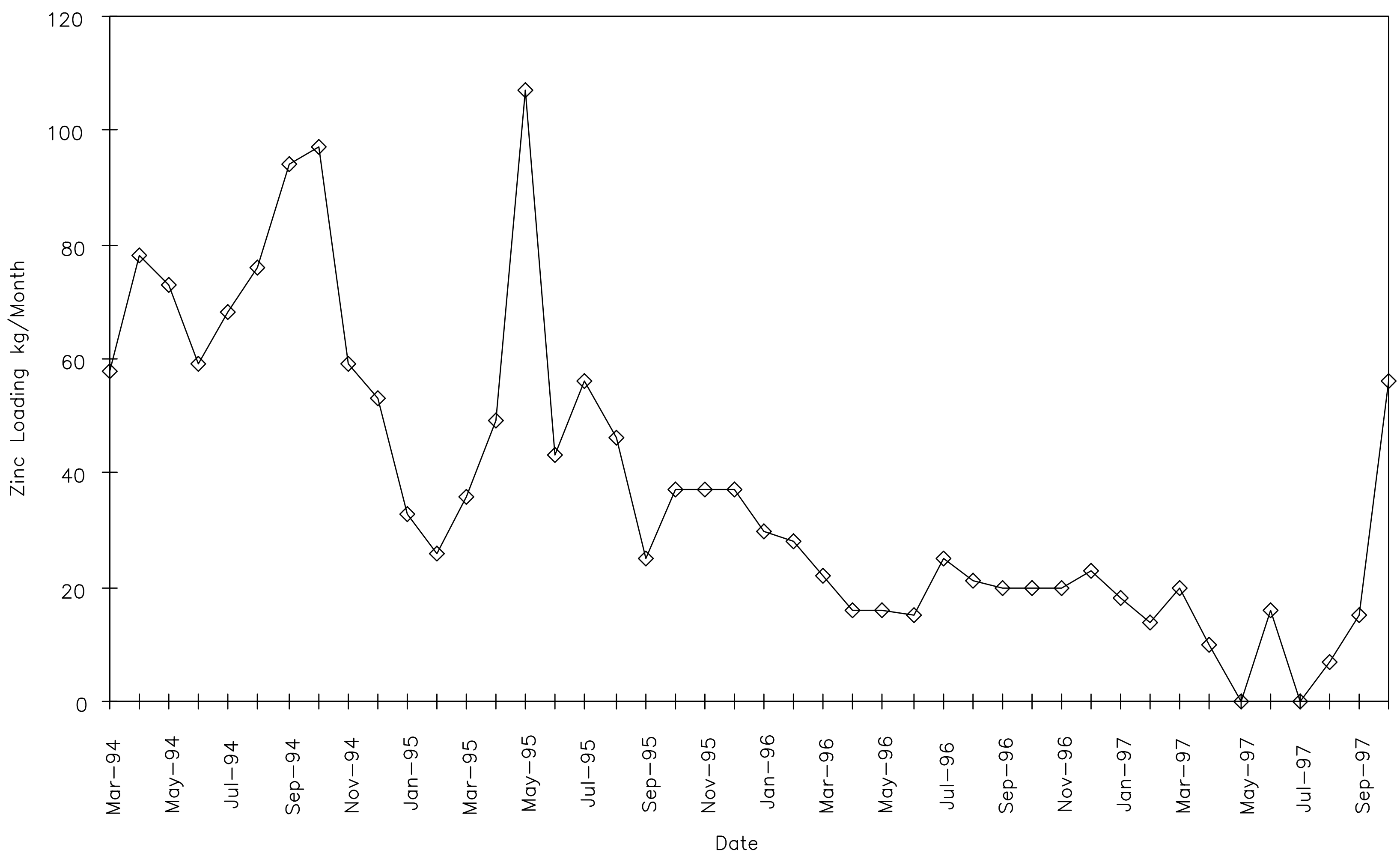

Figure 11. Monthly zinc loading, upflow CWS.

reduction must be great enough to reduce zinc concentrations in the substrate to below inhibitory levels. This hypothesis suggests that the effectiveness of an anaerobic compost CWS is a function of the rate of sulfate reduction, residence time of the mine drainage in the wetland substrate, and the concentration of zinc (or other inhibitory metals) in the mine drainage. Low temperature is also a factor that will affect the activity of sulfate-reducing bacteria in the wetland.

The following can be concluded from the evaluation of the upflow cell:

- The upflow CWS is effective in removing many metal contaminants from mine drainage; however, the CWS may have difficulty recovering from rapidly increasing metals loading conditions. Reinnoculation and incubation of sulfate-reducing bacteria may improve recovery of these systems.
- Control of mine drainage flow to the constructed wetland is critical to ensure that residence time and operational conditions are maintained.
- The operational lifetime of an upflow CWS (with a compost substrate depth of 4 feet) is roughly 4 to 5 years.
- The upflow cell had superior hydraulic performance throughout most of the demonstration.
- Winter freezing can be prevented by covering the wetland surface with hay or blankets used in curing concrete.
- Piping cleanouts should allow all piping networks to be easily cleaned.

3.4.4 Clear Creek

The untreated Burleigh Tunnel mine drainage and the effluents of both CWS cells discharge to Clear Creek. To assess the impact of treatment on the receiving stream, upstream and downstream samples collected from Clear Creek were also analyzed for total metals and aquatic toxicity. The metals results indicated that although the wetlands may be removing metals from the mine drainage, the demonstration-scale CWS treated only a small portion of the total discharge from the Burleigh Tunnel, not enough to show a measurable decrease in the metals content of the stream. The demonstration-scale CWS treated approximately 30 percent of the total flow from the Burleigh Tunnel, and during high flow treated only about 5 percent of the flow. A full-scale system could show a more significant decrease in the metals content of Clear Creek downstream of the system.

The stream results for upstream versus downstream samples are presented in Tables 6 and 7. The results show that Burleigh Tunnel mine drainage is a significant source of zinc to Clear Creek. However, CDPHE reports there are also additional nonpoint sources of zinc-contaminated water received by the creek.

3.4.5 Toxicity Testing Results

Constructed wetland treatment is a complex biogeochemical process involving adsorption, chemical precipitation, and microbial interactions with contaminants. The primary metal removal mechanisms in the CWS are chemical precipitation and microbial sulfate reduction; however, treatment may also complex metal contaminants, making them unavailable to receptor organisms. Thus, aquatic toxicity analyses were conducted by the EPA National Exposure Research Laboratory - Aquatic Toxicity during the demonstration to evaluate the reduction in toxicity resulting from CWS treatment. Two test organisms were used in the toxicity testing: water fleas (Ceriodaphnia dubia) and fathead minnows (Pimephales promelas). A total of eight rounds of aquatic toxicity testing were conducted during the demonstration. Initially, toxicity samples were collected and analyzed every 3 to 4 months until late 1995, when demonstration activities were temporarily suspended. When demonstration monitoring resumed, toxicity testing was conducted every 4 to 6 months. In 1997, a microbial toxicity test was conducted on wetland sulfate-reducing bacteria with Burleigh Tunnel mine drainage. The results of the microbial toxicity test are presented in Section 3.4.6.

Aquatic toxicity testing results correlated well with increasing zinc concentrations observed in the effluents of the treatment cells during the first 2 years of the demonstration. Results of testing conducted during the first 8 months of the demonstration indicate the effluents from both cells were not toxic to either the C. dubia or the P. promelas. The Burleigh Tunnel mine drainage was toxic to both test organisms at low concentration (dilution) throughout the demonstration. Table 8 provides influent and effluent concentrations resulting in the death of 50 percent of the test organisms (LC50) in each round of testing. As zinc concentrations increased in the effluents of both cells through 1995, so did the toxicity to the test organisms.

The first test conducted that year (February 1995) indicated that effluent from the upflow cell had become toxic to C. dubia at a concentration of 8.4 percent. The high runoff event that occurred in the spring of 1995 and

Table 6. Clear Creek Upstream

	Cadmium (mg/L)	Lead (mg/L)	Nickel (mg/L)	Zinc (mg/L)	pH	Conductivity (:S)	Temperature (°C)
Average	0.0022	0.0034	0.0047	0.126	7.8	155.7	5.4
Maximum	0.0094	0.013	0.015	0.56	8.1	167.5	9.7
Minimum	0.0	0.0	0.0	0.11	7.6	144.0	0.9

Notes:

°C Degrees Celsius
mg/L Milligrams per liter
:S MicroSiemens
ND Not Detected
pH Standard units
Average / Arithmetic Mean

Table 7. Clear Creek Downstream

	Cadmium (mg/L)	Lead (mg/L)	Nickel (mg/L)	Zinc (mg/L)	pH	Conductivity (:S)	Temperature (°C)
Average	0.00075	0.0013	0.0068	0.512	7.6	132.8	4.3
Maximum	0.0017	0.0024	0.026	0.56	8.1	173.3	9.7
Minimum	ND	ND	ND	0.14	6.5	80.0	--

Notes:

°C Degrees Celsius
mg/L Milligrams per liter
:S MicroSiemens
ND Not Detected
pH Standard units
Average / Arithmetic Mean

associated increases in flow through the CWS cells and elevated zinc concentrations resulted in higher zinc levels in the CWS effluents. At that time, the effluent from both cells became toxic to the test organisms. The upflow cell effluent was toxic to C. dubia at a concentration of 0.1 percent and to P. promelas at concentrations ranging from 1.2 to 2.3 percent. The downflow cell effluent was toxic to C. dubia at concentrations ranging from 0.31 to 0.51 percent and to P. promelas at concentrations ranging from 2.6 to 30 percent.

Over the final 2 years of the demonstration, the upflow cell effluent continued to be toxic to C. dubia at concentrations below 1 percent and to P. promelas at a concentration of 14 percent. Toxicity samples were not collected from the downflow cell: operation of this cell was discontinued in September 1996.

Demonstration toxicity testing results indicate that the ability of the wetlands to reduce toxicity to aquatic organisms gradually declined over the first 2 years. In addition, the high flow event in 1995 had a significant impact on zinc and toxicity removal by the upflow cell over the final 2 years of the demonstration.

Water samples for toxicity testing were collected from Clear Creek above and below the CWS discharge three times during the demonstration. As mentioned, the constructed wetlands treated only 30 percent of the mine drainage; thus, the impact of treatment on the receiving stream was minor. One set of samples contained higher toxicity in the upstream sample while samples collected after June 1995 indicated that there was no acute toxicity in the upstream samples but that addition of the mine drainage to the stream resulted in an increase in toxicity.

Table 8. CWS Demonstration Toxicity (LC_{50}) Results

Indicator Species	Date Collected	Influent	Upflow Effluent	Downflow Effluent	Clear Creek Upstream	Clear Creek Downstream
Fathead Minnows (*Pimephalus promelas*)	08/24/94	1.1	No toxicity	NA[2]	No toxicity	No toxicity
	09/19/94	0.73	No toxicity	No toxicity		
	02/22/95	1.6	No toxicity	No toxicity		
	06/12/95	1.0	2.3	2.6	No toxicity	No toxicity
	09/05/95	0.62	1.2	30		
	12/10/96	0.62	1.6	NA		
	06/24/97	0.69	24	NA	No toxicity	No toxicity
	10/29/97	1.4	14	NA		
	10/29/97[1]		11			
Water Fleas (*Ceriodaphnia dubia*)	08/24/94	0.46	No toxicity	NA	No toxicity	No toxicity
	09/19/94	0.31	No toxicity	No toxicity		
	02/22/95[1]	1.0	8.4	No toxicity		
	02/22/95			No toxicity		
	06/12/95	0.10	0.43	0.51	No toxicity	No toxicity
	12/10/96	0.09	0.22	NA		
	06/24/97	0.43	0.41	NA	No toxicity	No toxicity
	09/05/95	0.10	<0.19	0.31		
	10/29/97	0.15	0.13	NA		
	10/29/97[1]		0.19	NA		

Notes:

[1] Duplicate Sample
[2] NA - Not analyzed

3.4.6 Microbial Toxicity Testing

Microbial toxicity testing was undertaken when repairs to the upflow cell indicated that there were few metal sulfides in the wetland substrate compared with observations conducted in previous years. The lack of metal sulfide deposits in the substrate suggested that the sulfate-reducing bacteria were not actively producing sulfide. Thus, Burleigh Tunnel mine drainage was tested at the Colorado School of Mines for toxicity to sulfate-reducing bacteria isolated from the upflow cell.

The tests indicated that the mine drainage is inhibitory to sulfate-reducing bacteria at low concentrations (dilution) corresponding to a zinc concentration of 17.5 mg/L. In addition, zinc sulfate ($ZnSO4 \cdot 7\ H2O$) was used to show that the zinc was the toxic constituent (positive control) in the mine drainage. The zinc sulfate was also toxic to the sulfate-reducing bacteria at a similar zinc concentration (18.8 mg/L). The concentration of zinc in the Burleigh Tunnel mine drainage typically exceeds the inhibitory level measured in this study. A similar study conducted using Desulfovibrio desulfricans also found a zinc concentration of 13 mg/L resulted in inhibition to the bacteria. (Paulson and others 1997).

Evidence that sulfate reduction was important to the removal of zinc in the upflow CWS include the large population of sulfate-reducing bacteria observed when zinc removal was also high (first year of demonstration), the accumulation of AVS, primarily zinc sulfide, in the substrate of this cell, and the decline of sulfate-reducing bacteria populations after the high flow event that corresponded with lower zinc removal by the upflow cell.

Visible observations of the upflow cell substrate observed blackening of the substrate during the first year of operation suggesting metal sulfides were accumulating, however, observations of wetland substrate conducted three years later, showed little blackening of the substrate. These results suggest sulfate-reduction was not as an important metal removal mechanism and was occurring to a much lesser extent during the latter portion of the demonstration. These observations also suggest that previously formed metal sulfides are not stable when environmental conditions within the wetland changes.

3.5 Attainment of Demonstration Objectives

This section discusses the results of the CWS demonstration in regard to the attainment of primary and secondary demonstration objectives. In addition, metal removal mechanisms, some of the causes for poor performance, and substrate lifetimes are discussed for each cell.

The results of the demonstration were able to achieve many but not all of the primary objectives outlined in Section 3.3. The first primary objective was the measurement of wetland effectiveness with respect to cell flow configuration and seasonal variation. This primary objective was achieved in part. The demonstration zinc results indicate zinc removal is greater with an upflow configured wetland; however, the technology as tested is not capable of meeting low metal discharge requirements for extended periods.

The better zinc removal and flow of the mine drainage through the upflow CWS compared to the downflow CWS indicate the upflow configuration is superior. Unfortunately, it was not possible during this demonstration to determine the effect of season variation on the performance of the upflow CWS. The downflow CWS actually performed better during the winter. The reason for the improved winter performance is discussed in Section 3.4.2.

The second primary objective was to determine the toxicity of the Burleigh Tunnel mine drainage. This primary objective was achieved. The Burleigh Tunnel mine drainage is toxic to both the C. dubia and P. promelas. Measured LC50 values for the P. promelas (fathead minnows) ranged from 0.62 to 1.6 percent (mine drainage) and for the C. dubia (water fleas) ranged from 0.10 to 1.0 percent.

The third primary objective was the characterization of toxicity reduction resulting from CWS treatment. This primary objective was also achieved. The demonstration toxicity results indicate the ability of the wetlands to reduce toxicity to aquatic organisms declined over the first two years of operation. Further, the high flow event had a significant impact on toxicity removal in both wetland cells.

The final primary objective was to estimate the toxicity reduction to the mine drainage receiving stream (Clear Creek). This primary objective was not achieved as none of the demonstration stream samples were toxic to either test organism.

The most significant primary objective not achieved is the inability to determine the seasonal variability of the upflow CWS. During winter, constructed wetlands located in cold climates may be less effective as a result of lower microbial activity. This may require pretreatment of the mine drainage during winter, oversizing the CWS or retaining a portion of the flow until warmer conditions return.

The first secondary objective of the demonstration was to estimate the lifetime of the substrate material. The lifetime of substrate material is estimated to be 4 to 5 years. The estimate is based on the breakdown of the substrate material resulting in settling and compaction of the substrate that leads to flow restrictions. In addition, demonstration substrate data for nutrients indicate elements such as phosphate (orthophosphate) have been depleted in the substrate by this time. If low discharge limits must be met then demonstration results suggest the substrate lifetime is approximately one year (taking into account the demonstration starting time and freezing of the upflow cell during the first year). However, in this situation it would likely be more cost effective to pretreat the mine drainage or amend it with an electron donor such as ethanol to extend the lifetime of the substrate material.

The second, noncritical or secondary objective was to estimate metal removal by sulfate reducing bacterial. This evaluation was expected to be qualitative as the bacteria counts and acid-volatile sulfide analyses are not highly precise and the metal removal may not be uniform throughout the treatment cells. As discussed in Section 3.4.2, the downflow cell data did not indicate the primary metal removal mechanism to be sulfate reduction. Section 3.4.3 discusses the upflow cell results for sulfate-reducing bacteria removal of metals. Data indicated an initial high rate of removal with a longer term reduction in this mechanism of metals removal.

The third noncritical, secondary objective was to evaluate the impact of the systems effluent on Clear Creek. These data are discussed in Section 3.4.4, and indicate that although the treatment was effective in removing metals from the Burleigh Tunnel drainage, the relatively small portion of the discharge being treated did not produce a measureable decrease in the metals content of Clear Creek.

The fourth and final noncritical objective was to evaluate capital operating costs for the CWS. Section 5.0 of this report provides a detailed economic analysis and successfully provides data useful for estimating costs for application of this technology at other sites.

3.6 Design Effectiveness

The following sections discuss the effectiveness of the upflow and downflow CWS tested during the Burleigh Tunnel demonstration. The basic design of each wetland cell is discussed in Section 1.3.2 of this report. This discussion focuses on general design parameters and factors that affected each cell.

The basic design of the CWS demonstration system consisted of a dam inside the Burleigh Tunnel, piping from the dam to the influent weir, the two wetland cells, an effluent weir, and a bypass pipe. The dam collected the mine drainage and provided adequate hydraulic head to drive the mine drainage through the upflow cell. The influent weir partitioned the mine drainage to the CWS cells and channeled the excess water to the bypass piping. From the influent weir, the mine drainage was channeled to a ball valve that separated flow to the CWS cells. Water collected from the cells was piped to the effluent weir and was discharged to Clear Creek. The purpose of the effluent weir was to regulate flow through the wetland cells.

Construction materials associated with this design were generally inexpensive, readily available, and easily transported to remote areas. Installation techniques were also straightforward.

The major drawbacks of this design observed during the demonstration centered on the flow control valves and the inability of the effluent weir to regulate flow through the cells. Because flow through the cells could not be controlled with the effluent weir, flow through the cells was regulated at the influent weir and control valve. Unfortunately, this design meant that any adjustment in flow to one cell affected flow to the other cell. Future systems should use easily controlled flow structures such as weirs to regulate flow to both cells independently.

In addition, the capacity of the initial 4-inch bypass line was insufficient to accommodate the large water volume during spring runoff. Eventually, a 6-inch bypass line was installed. Piping connecting the influent control structure and the cells should be direct and accessible for routine cleanout.

A drawback associated with the use of compost substrates is the high concentration of nitrate in the effluent water during startup. During this demonstration, no attempt was made to remove the nitrate from the water prior to discharge. In a similar wetland evaluation, startup effluents were applied to surface soils. Alternatively, the startup effluent could be stored on site in a pond or tank and fed back into the CWS.

3.6.1 Downflow Cell

The downflow cell consisted of 4 feet of a compost (95 to 96 percent) and hay (4 to 5 percent) substrate. The mine drainage flowed from the top to a PVC piping collection network at the base of the cell. The influent and effluent distribution networks were staggered within the cell to minimize short-circuiting of the mine drainage in the substrate.

The design of the downflow cell is discussed in Section 1.3.2; Figure 2 shows a cross section of the anaerobic CWS in an upflow configuration. The downflow configuration is only a reversal of the influent and effluent flows, not the construction of the cell.

For the most part, the materials used in the construction of the cells–HDPE liner, geonets, and PVC piping were acceptable. However, the geofabric was found to fill with fine material and lose permeability over the 2½-year demonstration. In addition, the cell piping networks did not include cleanouts. Cleanouts should be included in future CWS designs. Finally, the influent piping network did not evenly distribute the mine drainage in this cell. An additional row of perforated piping in this cell would more evenly distribute the mine drainage.

The cell was designed to treat 7 gpm. However, during the demonstration, the downflow cell became less permeable. The permeability loss is believed to be related to precipitation of metal oxides, hydroxides, and carbonates, settling of fine materials in the cell, and compaction of the substrate material. In winter months, flow through the downflow cell improved; presumably, the contraction of frozen substrate allowed water to flow between the liner and the substrate. However, this short circuiting did not substantially affect metal removal by the cell.

In an attempt to restore flow through the downflow cell, air was injected into the substrate to fluff the material. Although this technique improved flow, the effect was typically short lived. The results of this demonstration indicate that substrates with high concentrations of compost will not retain permeability in a downflow configuration and are not recommended. However, some recent downflow wetlands have used substrate mixtures of 50 percent limestone with sawdust and compost to improve hydraulic characteristics.

3.6.2 Upflow Cell

The design of the upflow CWS is identical to the downflow cell except that the mine drainage is channeled up though the compost substrate. Figure 2 shows a cross section of the demonstration anaerobic compost CWS. The design of the demonstration wetlands is discussed in Section 1.3.2.

In general, the upflow cell retained permeability throughout the demonstration. However, some hydraulic restriction developed during the later half of the demonstration resulting in a preferential flow pathway. In addition, gas buildup produced by fermenative bacteria within the upflow cell may have restricted flow to the effluent lines in the wetland during the last year of the demonstration. Gas was released from the cell by periodically puncturing the upper geofabric with a pitch folk. Replacing the geofabric with a fine mesh geonet could eliminate gas buildup. Also, the decline of sulfate-reducing bacteria and apparent increases in the population of fermentative bacteria likely exacerbated the problem.

The upflow cell was prone to freezing during winter. During startup, the dike within the Burleigh Tunnel gave way, stopping flow to the upflow cell. Flow was restored by thawing the ice around the effluent line with a steam cleaner and water tank heater. The following winter, hay bales were placed over the substrate followed by insulated blankets (identical to insulated blankets used for curing concrete), and the system was operational throughout the winter. However, the straw bales became saturated with water and the combined weight compressed the substrate so that all flow ceased through the cell. Flow through the cell was restored once the hay bales were removed. During year three, the insulated blankets were used alone to insulate the cell and there were no interruptions in flow during this period. In the final year, the ponded water in the upflow cell was allowed to freeze and did so to a depth of approximately 6 inches. There were no interruptions in flow during that winter.

Residence time is an important factor in anaerobic constructed wetlands that use sulfate-reducing bacteria. Decreasing residence times may overload the wetland, exposing the bacteria to inhibitory concentrations of zinc. Based on the size of the wetlands and substrate water volumes (percent moisture results of 50 percent) the calculated residence time for a flow rate of 7 gpm is 48 hours, and 67 hours at a flow rate of 5 gpm. Verification of residence times was one of the more difficult measurements undertaken during the demonstration. Both a chloride tracer (treatability study) and an organic dye test (demonstration) were unsuccessful in measuring residence time. The chloride could not be readily measured as background levels of dissolved salts was somewhat high during the treatability study and the organic dye likely absorbed to the wetland substrate during this demonstration test.

During the final year of the demonstration, flow through the upflow cell began to short circuit in an area adjacent to the southeastern bermed sidewall. An excavation was made into the wetland to the influent line feeding this section of the cell and the line was capped. Dewatering the excavation was somewhat difficult and would have been aided by a sump within the cell. Inspection of the influent line found precipitates coating the piping walls and in the piping perforations. The amount of material in the perforations and the pressure on the piping against the geofabric would have caused a notable restriction in flow. Replacing the geofabric with a fine mesh geonet should alleviate the problem.